A REVIEW OF RADIOACTIVE WASTE DISPOSAL
TO THE GROUND AT HANFORD

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Principality of Monaco

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Hanford Laboratories
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General Electric Company
Richland, Washington

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INTRODUCTION

The philosophy, techniques, and controls applied at Hanford in the ground disposal of certain radioactive wastes were described at the first Geneva Conference on the Peaceful Uses of Atomic Energy (1). At the second Conference, disposal facilities were discussed, volumes and contained curies were reported, and experiences were highlighted (2). The present paper will summarize ground disposal (3), review significant laboratory and field soil science and mineralogical research (4) with emphasis on recent findings (5), and describe the extent of groundwater contamination around the disposal sites (3).

OPERATING EXPERIENCE

The separations plants at Hanford discharge large volumes of solutions containing radioactive materials to various ground disposal facilities. These facilities, together with the volume and contained curies discharged through mid-year 1959 are schematically shown in Figure 1; the geology and hydrology of the site (4, 6, 7) are also indicated.

The largest contribution to the waste volume is made by the process cooling water streams. These discharge to natural depressions in the ground surface which are locally called "swamps". This waste is normally uncontaminated but infrequently contains low-level contamination due to corrosion failure of process equipment. The present recommended maximum limit for radioisotopes in wastes sent to swamps is 5×10^{-5} uc beta emitters/cc. Through June, 1959, a total of 139,000,000,000 liters and about 2500 fission product curies have been discharged to swamp sites.

Trenches are single-use facilities into which high salt wastes or wastes containing chemically complexed radioisotopes are discharged. The effectiveness of the disposal depends upon the specific retention properties of the some hundred meters of dry calcareous clays, sands, and gravels existing at the sites. A description of a waste that has been discharged to a trench is helpful in understanding this disposal technique.

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HANFORD SITE CROSS SECTION SCHEMATIC
SHOWING WASTE DISPOSAL FACILITIES

SWAMP (3)
13.9 x 10^16 LITERS
2500 c.FP

TRENCH (18)
10.6 x 10^7 LITERS
6.47 x 10^5 c.FP

CRIB (72)
14.9 x 10^9 LITERS
1.9 x 10^6 c.FP

BOILING TANK

WELL
547 DRILLED TOTALING 107,000 FT.
74 TO BASALT
322 TO GROUND WATER
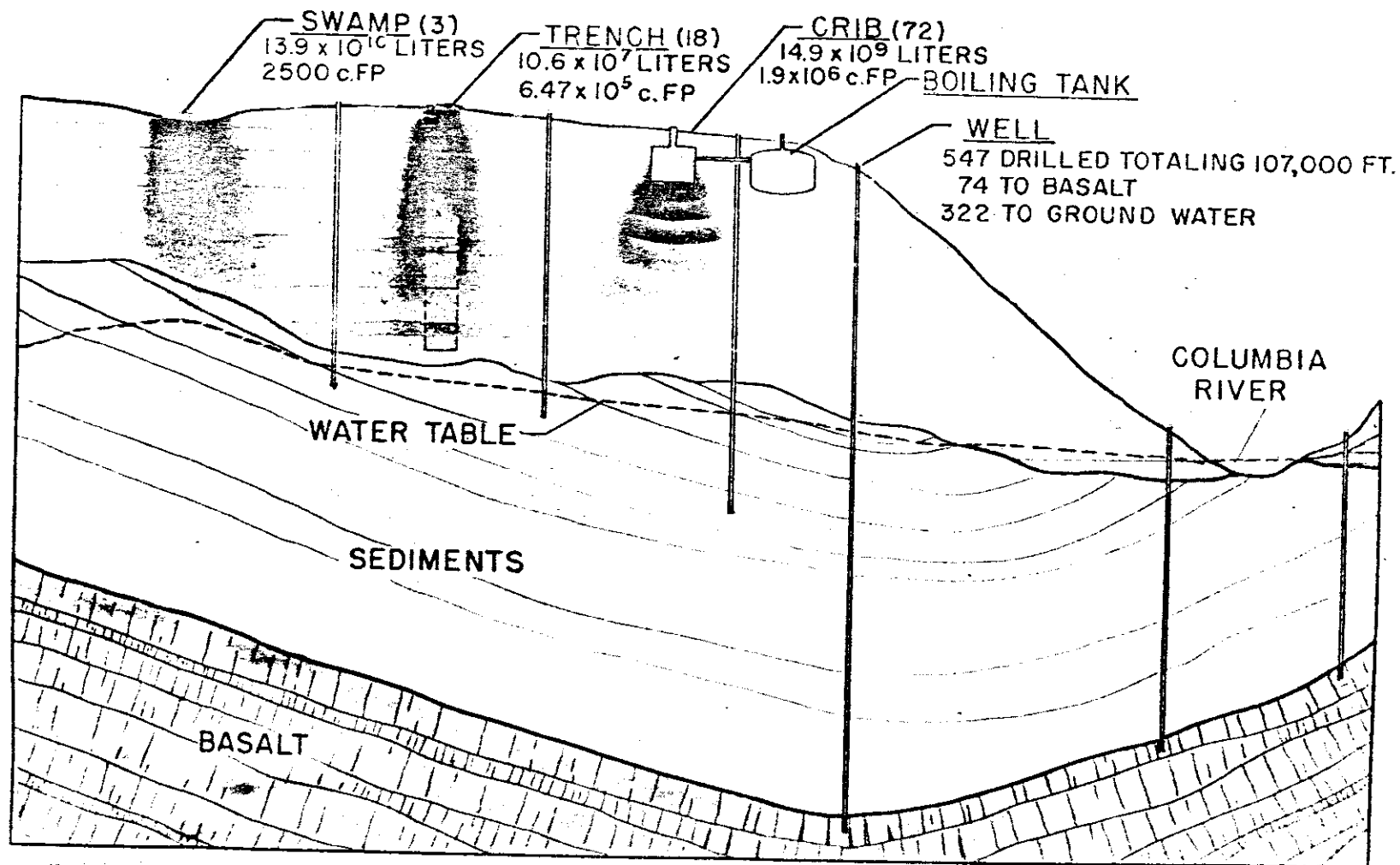
WATER TABLE

COLUMBIA
RIVER

SEDIMENTS

BASALT

FIGURE 1

The first Hanford separations plants produced a high activity-level solution containing the depleted uranium and the fission products; this was placed in underground storage tanks. The uranium was later recovered from this solution and a mixed fission product solution of somewhat greater volume remained for either tank storage or ground disposal. A co-precipitating or "scavenging" process was developed (8) to remove from this fission product solution the majority of the Sr^{90} and Cs^{137} which was returned to the tanks as a sludge. Analysis of the solution which had been separated from the sludge revealed the presence of Co^{60} presumably arising from the activation of natural cobalt impurity in the fuel elements. The chemistry of the scavenging process rendered this radioisotope almost incapable of ion exchange in the soil, probably due to the formation of complex ions. While some of this solution was disposable to the ground through the usual cribbing procedures (9), it was necessary to dispose of most of it by trenching which does not rely on adsorption reactions as a major holding mechanism. Through June, 1959, 18 trench sites have received 106,000,000 liters of waste containing 647,000 curies of fission products.

Process condensate wastes, which generally contain radioisotopes in concentrations of 10^{-1} to 10^{-3} uc gross beta emitters/cc, and those waters which have the potential of becoming highly contaminated are discharged to sub-surface cribs. The classical crib design was a box-like, timber structure; more recent types have trapezoidal cross-sections and may vary in length from 6 - 500 meters. They are filled with washed and sized gravels to promote even distribution of the waste solution. The wastes are piped into them underground and from them the wastes percolate through the soils toward the water table. The effectiveness of this disposal method depends upon adsorption and other reactions of the radioisotopes with the soil. Through mid-1959 seventy-two crib structures have received 14,900,000,000 liters containing 1,900,000 curies of fission products.

The term "gross beta emitters" requires some clarification. Generally it may be used as a measure of the fission product mixture of normal composition range as determined primarily by irradiation time in the reactors and cooling time prior to analysis. This normal range is distorted in certain waste streams to a significant degree, however, by chemical processing which segregates certain of the fission product elements into different streams. The scavenging process, for example, removed important quantities of Sr^{90} and Cs^{137} and these were returned to storage in the high level waste tanks; the remaining solution was therefore depleted in strontium and cesium and, for this reason, the gross beta measurement represents a mixture more largely ruthenium and much less hazardous than a normal fission product mixture. For another example, the vapors from tanks of self-boiling waste have a fission product content differing greatly from the normal; the

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condensates from these vapors, which are sent to cribs, contain a smaller percentage of Sr^{90} owing to its precipitation and retention in the tanks. Again the gross beta measurement represents a mixture differing greatly from the normal. Analyses for individual radio-isotopes in the wastes or in the ground water samples are made when required.

Of the total fission products discharged to the ground it is estimated that 75 percent has decayed to non-radioactive nuclides.

SOIL SCIENCE

Wastes discharged to the ground filter through the soils toward the water table. Ion-exchange and other reactions with soil minerals are utilized to retain fission products in the soil column. Figure 2 pictorially indicates the results of this exchange process. As the wastes flow downward, the strontium, cesium, and other fission products which have favorable adsorption properties are removed by the soil in an order determined by the characteristics of the soil and of the solution concerned. Ruthenium in processed solutions does not have good adsorption characteristics and continues downward to enter the ground water. The illustration of these chemical and waste flow effects is highly idealized and simplified to a major extent. The shape of the "cone" below the disposal facility is in actuality largely unknown; the influence of strata of different permeability, of saturated or near-saturated flow down the axis and unsaturated flow toward the edges of the body are as yet little understood. In particular, the illustration assumes homogeneous soils, idealized and unchanging waste concentrations, and uniform pH and temperature; none of these is obtained in actual practice.

Radioisotopes entering the ground water are looked for in samples from perforated steel-cased monitoring wells drilled around the facility and down the water table gradient. Weekly samples are taken from wells monitoring active disposal facilities; less frequent samples are taken from wells monitoring abandoned facilities.

Present techniques in the disposal of low and intermediate level wastes to the ground at Hanford have been developed through many laboratory and field experiments. Separate laboratory soil chemistry studies have been made with Sr^{90} (5, 10, 11, 12), Cs^{137} (12, 13), Pu^{239} (14), Ce^{144}, Y^{90}, and others (12, 15). Many of the earlier studies were of the equilibrium type in which a 1:10 soil-to-solution mixture was shaken overnight then centrifuged and the supernatant solution analyzed for the ion of interest. A distribution coefficient or K_d was then calculated as:

$$K_d = \frac{Counts/min/g. \ soil}{Counts/min/ml. \ solution}$$
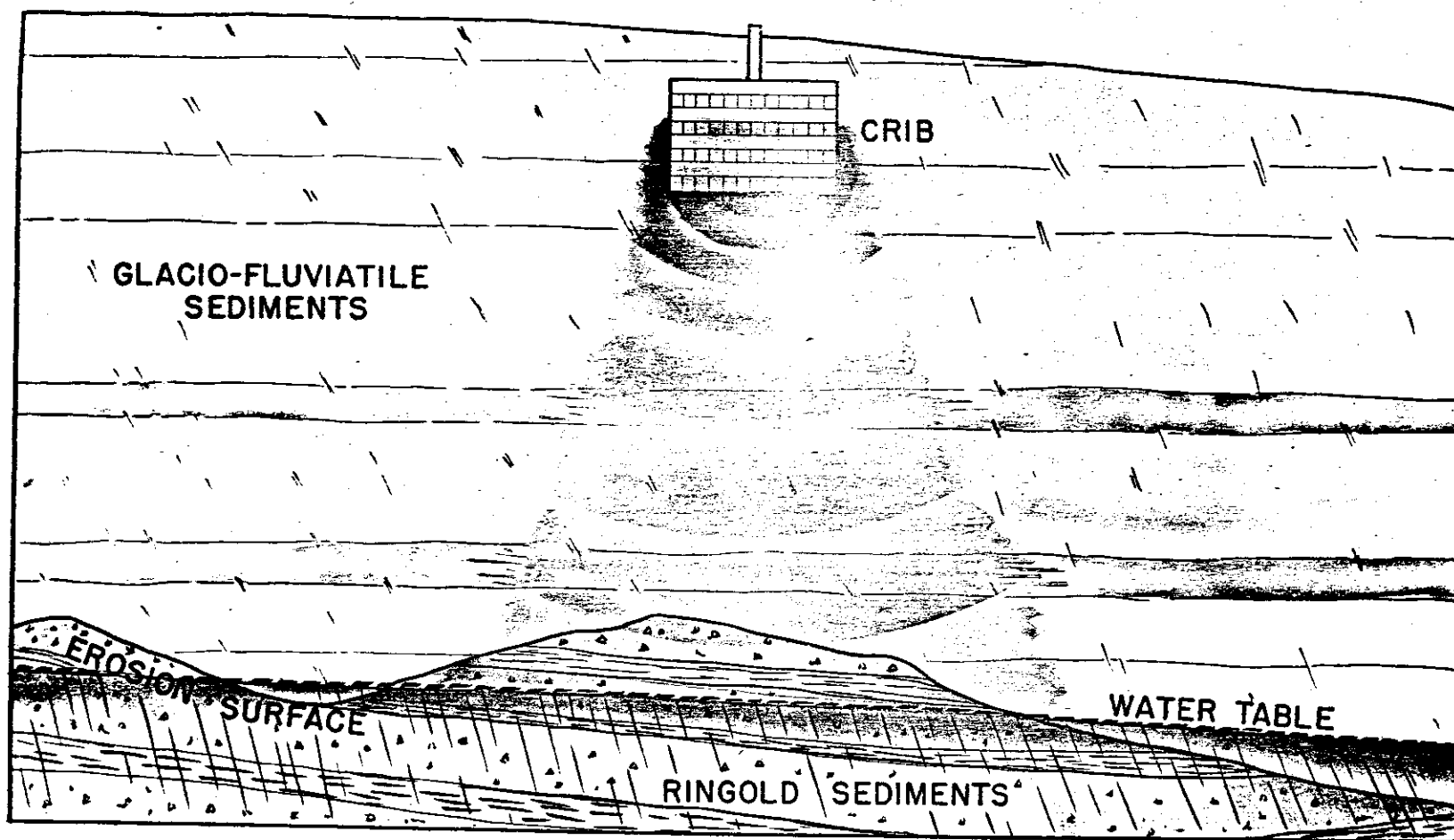
CROSS SECTION BENEATH A CRIB

FIGURE 2

More recent work has been done largely with soil columns in which a
solution containing the ion of interest was fed at a constant rate
to one end of a column and the effluent collected with a fraction
collector from the other. The fractions were then analyzed and a
breakthrough curve drawn plotting $\dfrac{\text{concentration effluent}}{\text{concentration influent}}$ against
the number of column volumes of solution fed to the column.

McHenry (10, 11) found that strontium was adsorbed by soil as a divalent
cation according to the law of mass action. Factors influencing sorp-
tion were found to be Sr concentration, pH, time of contact, nature and
concentration of complementary ions, and total salt concentration. The
influence of $PO_4^=$ on Sr uptake in the presence of $CaCO_3$ and related
mineral structures was found to be anomalous; the reaction will be con-
sidered later.

Recent soil column and equilibrium experiments verify the effect of pH
on Sr sorption in the presence of gross quantities of foreign ions.
For example, the influence of pH on Sr sorption in the presence of 2000
ppm of Na has been studied. At pH 3, about 10 column volumes were re-
quired for 5% breakthrough of Sr and K_d was 7. At pH 11, more than 75
column volumes were required and K_d was about 25.

Soil column work with actual wastes and operating experience have in-
dicated that the capacity of Hanford soil for sorption and retention of
most radionuclides is generally very greatly increased by neutralization;
most wastes have therefore been neutralized before release to the ground.

Soil column studies (5) have also shown that flow rate and temperature
have an influence on Sr sorption by soil. Where Sr is the only cation
present, no temperature effect is seen but, when Sr must compete with
other cations for exchange sites, a considerable effect may be observed.
Figure 3 illustrates the decreased soil uptake of Sr relative to that of
Mg at elevated temperatures from a solution containing 1000 ppm Mg and
0.1 ppm Sr. The effect is presumably due to dehydration of ions at elevated
temperatures. In the Mg-Sr system, illustrated in the figure, Mg ions
have more water of hydration than Sr ions. Ions that are normally highly
hydrated lose much of their water at increased temperature and approach
closer to the center of charge thereby competing more effectively for the
exchange sites. In the case of any particular solution then, the ion
of interest may be adsorbed to a greater or lesser extent at elevated
temperatures depending on whether the foreign ions present are hydrated
more or less than the ion of interest.

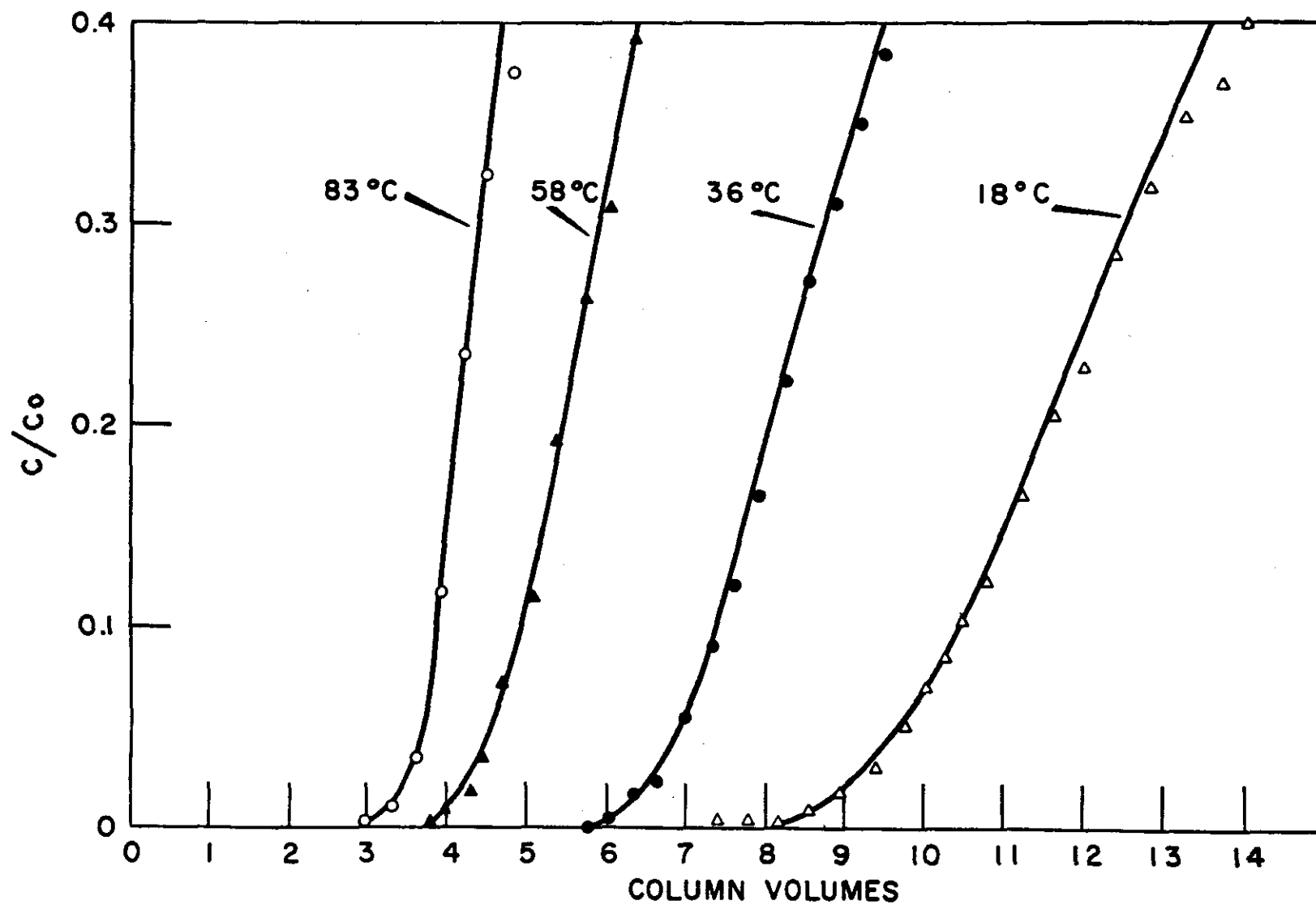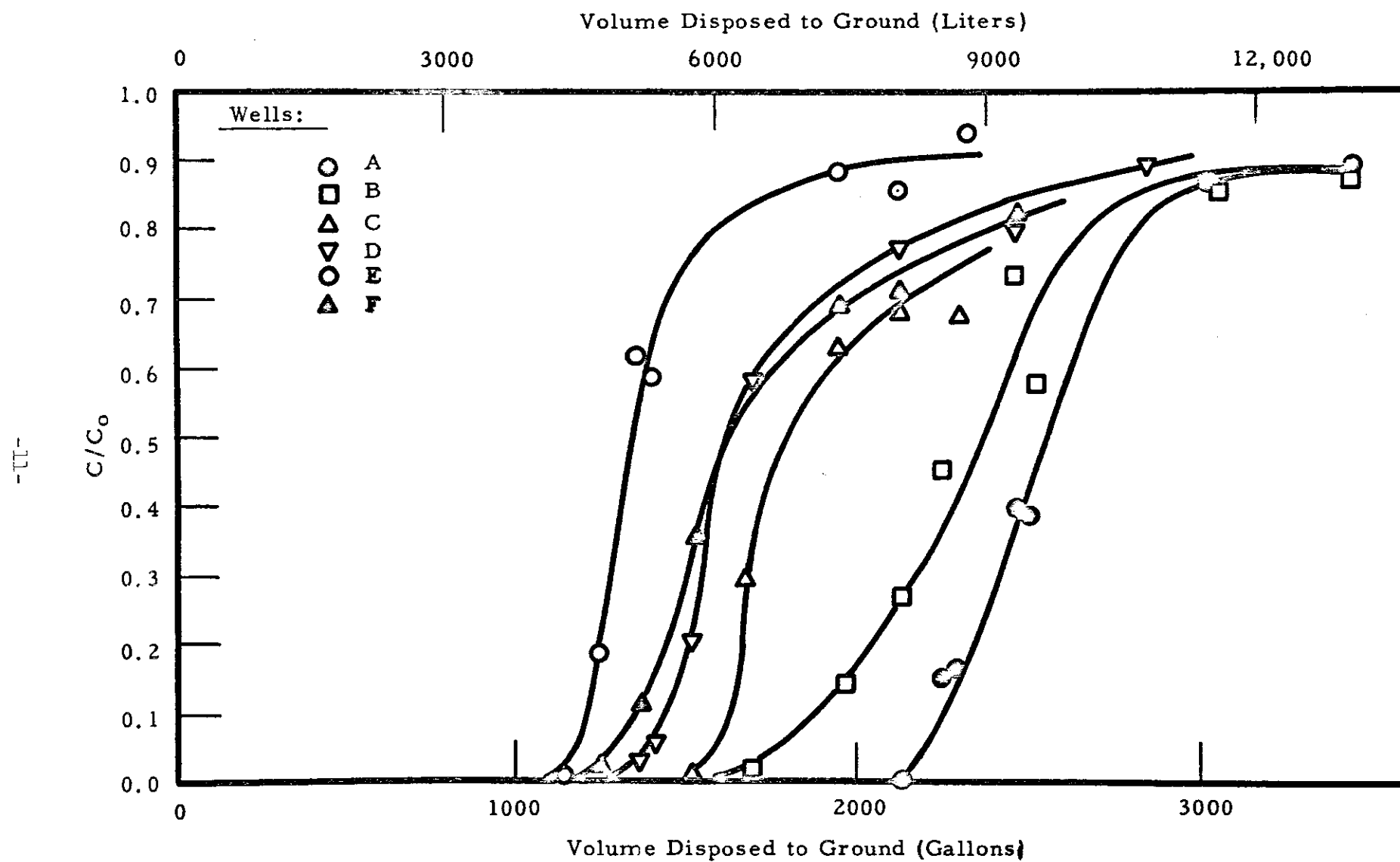# BREAKTHROUGH OF Sr COMPETING WITH Mg ON HANFORD SAND AT VARIOUS TEMPERATURES

FIGURE 3

In order to predict breakthrough of nuclides into ground water suffi-
ciently far in advance to have new disposal facilities available when
needed, laboratory soil column and equilibrium tests are made for each
disposal facility with each waste of concern. Extrapolation from short
(40 cm) columns to 60 to 100 meter field columns has been done on the
assumption that the same breakthrough curve will apply. The "column
volumes" plotted on the abscissa are calculated for field application
by assuming no lateral spread from the edges of the crib.

Recent experiments to determine more precisely the effect of column
length on breakthrough curves have been made with laboratory column
lengths up to 15 meters. These experiments indicate that the slope of
breakthrough curves becomes greater with greater column length. Thus
data from short columns predict initial nuclide breakthrough sooner
than it actually occurs in long columns and this, in turn, indicates
that predictions have been conservative from this standpoint.

In order to investigate more fully the applicability of laboratory
findings, a field experiment with a model crib is underway (17). The
model is a bottomless box 60 x 60 x 60 cm set in the ground in a loca-
tion where ground water is only about 3.6 m below the ground surface
and the overlying soil is fine sand, apparently homogeneous. Seven
well casings, 5 cm in diameter and screened on the bottom third of their
length, were set one meter into ground water in a pattern around the
crib as shown in Figure 4. Well F was driven at an angle such that
ground water was intercepted directly beneath the crib. Contours on
the water table map indicate that ground water movement is toward Well G.
Influent to the crib contained 600 ppm Ca as $Ca(NO_3)_2$ and about 0.1 ppm
Sr traced with Sr^{85}. Float valves admitted influent at a rate such as
to keep a constant level of solution in the crib. Samples of ground water
were taken from the wells at close intervals and a gamma detecting probe
was used to follow activity near the well casings above ground water.

In this first experiment with the model crib, nitrate was detected
first in the ground water in Well E (Figure 5) rather than in Well F.
Downward movement of the solution was therefore not completely verti-
cal but was displaced, probably because of soil heterogeneity. Nitrate
ion concentration in the well samples ultimately reached about 90%
of the influent concentration presumably because of dilution with ground
water. Nitrate ion is an adequate tracer of water movement through these
soils and the order of appearance of this ion in the various wells re-
flects the lateral distribution of liquids beneath the crib. In Well E,
50% of the ultimate concentration was reached after 5000 liters of solu-
tion had been charged to the crib. Using the known soil porosity of 31%,
it may be inferred that the solution above the water table occupied a
volume at least 16 times as great as would be calculated from the volume
directly beneath the crib assuming no lateral spread.

To Pond

G

D

C   45°

B   45°   F

E

12'

4'

Crib

2'

A

Experimental Crib Showing Well Placement

FIGURE 4

EXPERIMENTAL CRIB NITRATE BREAKTHROUGH CURVES

These findings indicate that spreading beneath a ground disposal site may be of greater benefit in waste disposal than has previously been assumed. It is expected that an understanding of scale-up procedures will be obtained through further work with the model.

Breakthrough curves for Sr[85] (Figure 6) show that Sr was found first in Well F directly beneath the crib. A 50% Sr[85] breakthrough was achieved in this well at 23,500 liters.
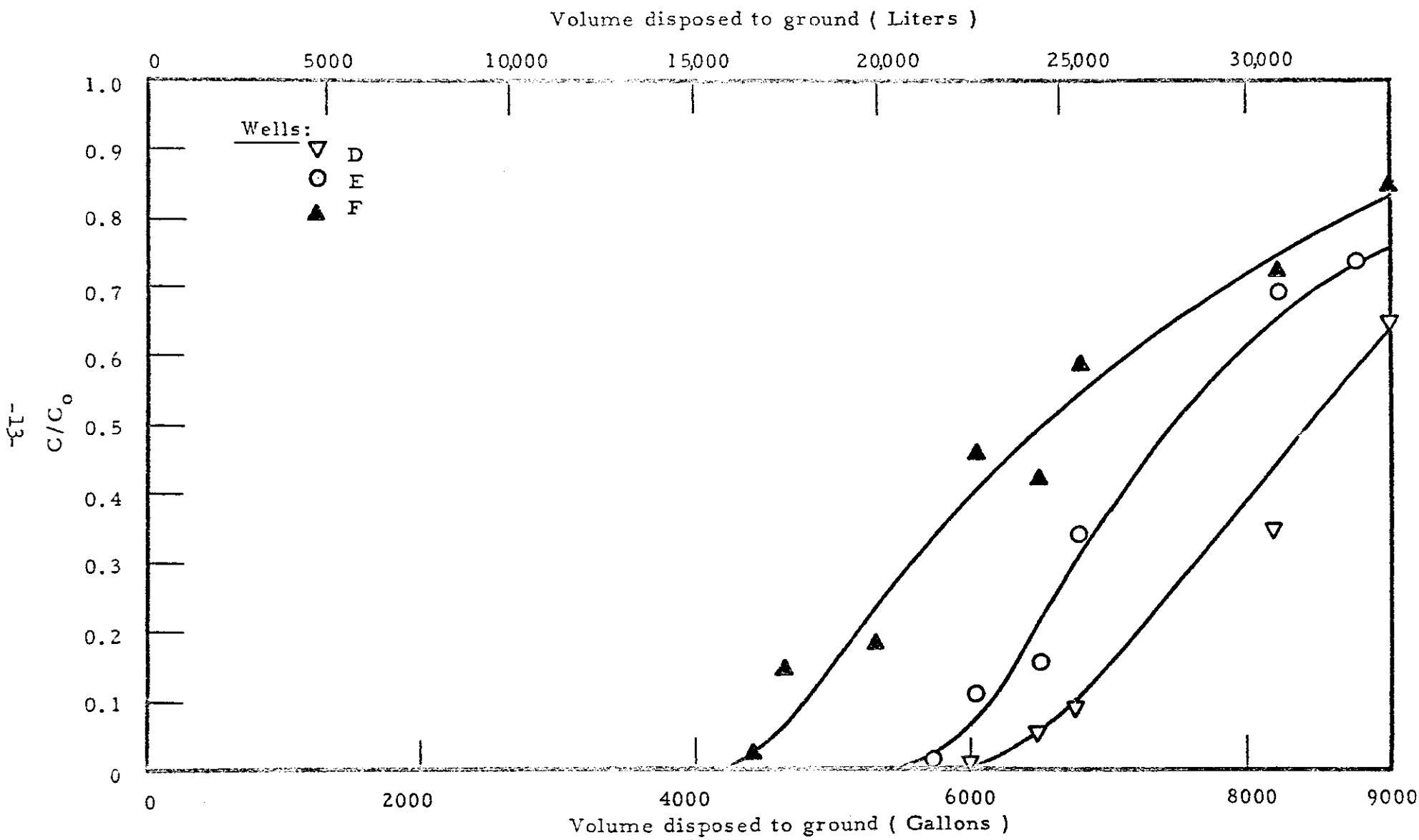
MINERAL REACTIONS

During the course of soil studies at Hanford, it was noted that the addition of phosphate to some waste streams improved the soil uptake of Sr^{+2} (18). It was found later that this was due to a calcite-phosphate reaction in which the mineral apatite was formed by a "replacement" process from the calcite. During the formation of the apatite some of the lattice positions normally occupied by Ca^{+2} ions were filled by Sr^{+2} ions in a new crystal (19). The equation for the reaction is

$$3PO_4^{-3} + \underline{5CaCO_3} + OH^- \rightleftharpoons \underline{Ca_5(PO_4)_3OH} + \underline{5CO_3^{-2}}$$
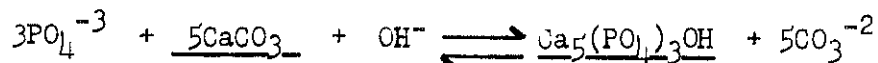
Undoubtedly the reaction plays a significant role in the removal of Sr from phosphate-containing wastes by the calcareous soils of the Hanford site.

More recently, columns of crushed calcite have been studied quite extensively for possible application in waste treatment processes. The significant variables which affect Sr^{+2} removal have been found to be calcite surface area, pH of the influent solution, column flow rate, temperature, and phosphate ion concentration. Large concentrations of alkali metal ions or aluminate ions do not interfere, although alkaline earth metal ions may cause the formation of precipitates upon the addition of phosphate.

Table 1 shows the relative effect of various calcite particle sizes. Sr^{+2} is removed more rapidly where surface area is greatest.

Volume disposed to ground (Liters)

Wells:
▽ D
○ E
▲ F

C/C$_o$

Volume disposed to ground (Gallons)

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Experimental Crib Sr^{85} Breakthrough Curves

Table 1

Relative Amounts of Strontium Removed from Phosphate Solution by Calcite of Various Grain Size Ranges

Calcite Grain Size Range, mm	Removal of Strontium, %
1.0 to 2.0	33.9
0.25 to 1.0	36.8
0.05 to 0.25	69.2
<0.05	80.0

Solution-calcite contact time, hr   1.0
Calcite weight, mg   400
Initial solution pH   11.1
Initial solution: $3M$ $NaNO_3$; $0.05M$ Na_3PO_4; 2 mg $Sr(NO_3)_2/l$;
       100,000 d $Sr^{90}-Y90/m/ml$; 40 ml solution/
       sample

Figure 7 shows the effect of pH on Sr^{+2} removal by calcite. It can be seen that high pH values favor the reaction. Below pH 7 calcite is soluble and the reaction with phosphate to produce apatite does not occur. This may be a major disadvantage of the calcite-phosphate reaction for waste stream decontamination since in many waste streams ions are present that precipitate on neutralization or the addition of phosphate. Sedimentation, filtration or centrifugation would then be necessary before passage of the solution through a calcite column.

Figure 8 shows the influence of column flow rate and temperature on the removal of Sr^{+2} from solution. Flow rate must be kept quite low to obtain a high decontamination factor; decontamination factors as high as 10^6 can be obtained with flow rates of 10 $ml/cm^2/hr$.

Temperature also has an appreciable effect on Sr removal, especially at high flow rates. At low flow rates the curves of Figure 8 approach a common point and the temperature effect approaches zero.

The concentration of phosphate in solution is important and maximum Sr^{+2} removal is approached at about $0.05M$ PO_4^{-3}. Figures 9 and 10 show the effect on Pm^{+3} removal of increasing PO_4^{-3} concentration from $0.005M$ to $0.05M$ (20). The figures also show that other cations, predominantly those of the "bone-seeking" group, are captured by the growing apatite lattice while monovalent ions such as Cs^{137} are not removed to an appreciable degree by this reaction.

Other natural carbonates of related structure, such as siderite or rhodochrosite, can also be used as effectively as calcite for removal of the "bone-seekers". Some of these other minerals may be of interest for radioisotope recovery because subsequent chemical separations would be easier in some cases.
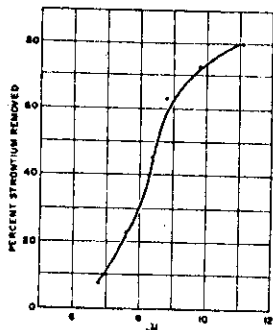
Figure 7

Variation of Sr^{+2} Removal by Calcite with Influent Solution pH from a Closed System. Initial solution: $0.05M$ Na_3PO_4; 10,000 d $Sr^{90}-Y^{90}/m/ml$; 500 mg, 0.077 mm calcite. Contact time: 1.0 hr.
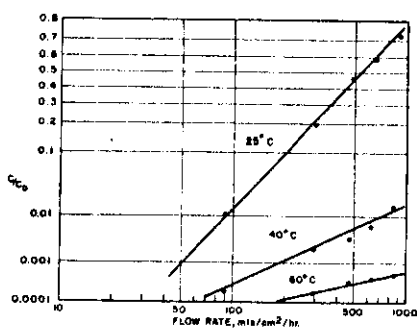

Figure 8

Variation of Sr^{90} Removal from a Kinetic System with Flow Rate and Temperature. Influent solution: $0.05M$ Na_3PO_4; $1M$ $NaNO_3$; 1.5×10^{-4} mg Sr/l; pH = 11.9 Column: 50g, 0.25 to 1.0 mm calcite.
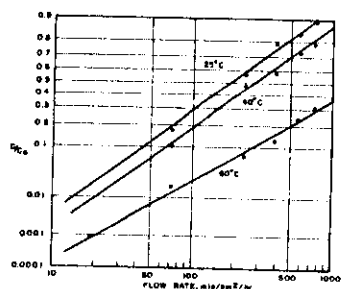

Figure 9

Variation of Pm^{147} Removal from a Kinetic System with Flow Rate and Temperature. Influent solution: $0.005M$ Na_3PO_4; 2.87×10^{-5} mg Pm^{+3}/l; pH = 11.8 Column: 50g, 0.25 to 1.0 mm calcite.
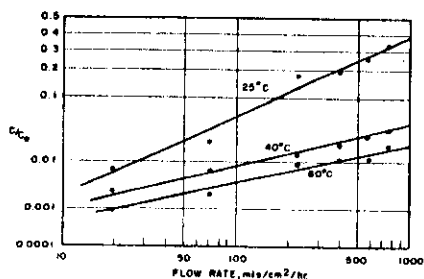

Figure 10

Variation of Pm^{147} Removal from a Kinetic System with Flow Rate and Temperature. Influent solution: $0.05M$ Na_3PO_4; 2.87×10^{-5} mg Pm^{+3}/l; pH = 11.8 Column: 50g, 0.25 to 1.0 mm calcite.

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Other mineral reactions of the replacement type have been studied, including the calcite-fluoride reaction (21). It was found that a reaction between F^- and calcite with neutral to alkaline solutions containing U, Sr, Pu, or rare earths, resulted in the removal of these radioisotopes from solution. Ideally the reaction is

$$CaCO_3 \; + \; 2F^- \; \longleftrightarrow \; CaF_2 \; + \; CO_3^{-2}$$
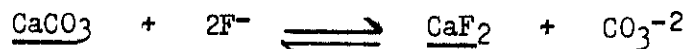
During the recrystallization and replacement process, cations with suitable size and charge enter the fluorite lattice. Again, cations removed into the fluorite are predominantly the "bone-seekers".

A search was made for a similar replacement reaction for application to the Cs case. None has been found to date. Several natural zeolites have been studied, however, for the removal of Cs from solutions; clinoptilolite ($Na_2O \cdot Al_2O_3 \cdot 8SiO_2 \cdot 6H_2O$) showed the greatest Cs specificity.

Figure 11 shows the effect of 1 N solutions of the cations indicated in competition with $0.01N$ Cs^+, $1.74 \times 10^{-8}N$ Cs^{137}.

Figure 12 shows the high Cs selectivity of clinoptilolite with competing Na^+ in various concentrations. Above $3N$ Na^+, the effect of increasing the Na^+ concentration is slight.

Clinoptilolite exhibits a high specificity for Cs^+ and has an exchange capacity greater than that of any of the common clay minerals studied such as illite, montmorillonite and vermiculite. The material can be crushed and screened to obtain a desired particle size, while fine-grained clays must be bonded and extruded for use in columns. Clinoptilolite is much more acid-stable than clays, or even than most other zeolites. The increased thermal and radiation stability of the inorganic clinoptilolite over that of organic cation exchangers is another practical advantage.

More than 13,000 column volumes (63 ml/column volume) of tap water containing 200 ppm salts, chiefly calcium and sodium carbonate, and 1×10^{-2} uc Cs^{137}/ml were passed through a 50g, 0.25 to 1.0mm clinoptilolite column at a flow rate of 1590 ml/cm^2/hr without detectable Cs^{137} breakthrough; analytical sensitivity was ca. 10^{-5} uc/ml. Decontamination factors of 10^3 to 10^4 were obtained.

To compare this performance with that of the replacement reactions, a 50g column of 0.25 to 1.0mm calcite (1.9 cm diameter x 12 cm long) was studied. With $0.05M$ Na_3PO_4 and 1×10^{-2} uc Sr^{90}/ml, Sr breakthrough was first detected (analytical sensitivity, 7×10^{-8} uc/ml) at 2800 column volumes when Sr^{90} decontamination factors began to fluctuate between 10^5 and 10^6 at flow rates of 10 to 20 ml/cm^2/hr. At 3300 column volumes the decontamination factor had decreased to 10^3.

Figure 11
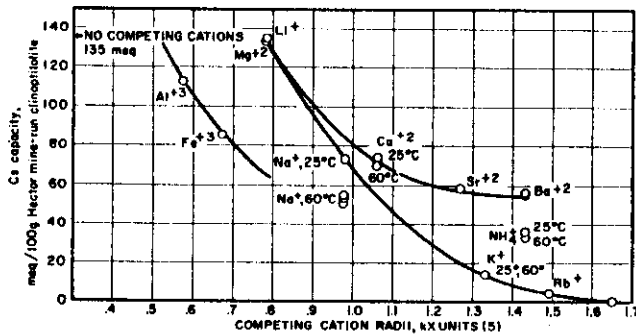
The Effect of Competing Cations on the Cs Capacity of Clinoptilolite.
Common influent solution: 1 Normal competing cations as indicated: 0.01N Cs^+; 1.74 x 10^{-8}N Cs^{137}
Common temperature: 25°C unless otherwise indicated
Common influent pH: 3.0, except 1.2 for Al^{+3} and Fe^{+3}
Common flow rate: 294 ml/cm²/ hr.
Common column: 50g, 0.25 to 1.0 mm, Na-based clinoptilolite.

Figure 12
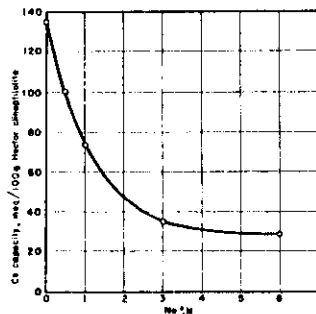
The Effect of Na Concentration on Clinoptilolite Cs Capacity.
Common influent solution: Na concentration as indicated: 0.01N Cs^+; 1.74 x 10^{-8}N Cs^{137}
Common temperature: 25°C
Common influent pH: 3.0
Common flow rate: 294 ml/cm²/ hr.
Common column: 50g, 0.25 to 1.0 mm, Na-based clinoptilolite.

The calcite reactions will occur when phosphate- or fluoride-containing
wastes are disposed to calcareous soils. It is believed that these
and other mineral reactions will be of value additionally in the treat-
ment of radioactive wastes prior to environmental disposal.

THE CONTAMINATION PICTURE

For the purposes of waste management, knowledge of the adsorption pro-
perties of the soil and of the ground water contamination is required
for assuring the continued safe disposal of radioactive wastes; knowledge
of the exact location of the radioisotopes fixed on a given soil column
is not considered to be essential. A field study was performed, however,
to study the distribution of radioisotopes on the soils beneath one
crib site. Early in 1956 a moderate well drilling program was carried
out around an abandoned crib which had received 151,000,000 liters of
predominantly acid waste containing 753,100 gross beta curies of fission
products. The original wells drilled prior to using the crib were
deepened and an additional twelve wells were drilled at various distances
from the crib center. Soil samples were taken for analysis during
the drilling. Figure 13 shows pictorially some of the results of this
study. The wastes spread from an initial distribution area of about
370 m^2 to an area of approximately 7400 m^2 at the water table. The
shaded areas indicate the concentrations of the beta emitters in the soil.
The highest concentrations were found in the soils immediately below
the crib. Figure 14 is another section of the soils below the crib
showing the probable distribution of the radioisotopes Cs^{137}, Sr^{90} and
Ru, as determined from the analytical data. Since Cs^{137} was not found
in any of the wells drilled after the crib was abandoned, although Ru
and Sr were accounted for, it is assumed that Cs^{137} had been retained
in the soils immediately below the crib. This is in good accord with
the knowledge that Cs has decreased mobility in comparison with Sr
when acidic wastes are passed through soil. Sr^{90} was largely contained
in the upper soil layers; however, some did penetrate to the ground
water in the vicinity of the W-22-15 well. Ruthenium, which is the
major ground water contaminant, was found to be spread more widely than
any of the other radioisotopes as expected. The eastward trend of the
contaminated soil body is explained by the structure of the beds;
easterly dipping foreset deltaic bedding is general in the Hanford flu-
viatile sediments and has been detected and measured in the vicinity of
this particular site.

The large volumes of wastes discharged to ground at Hanford have resulted
in a certain degree of ground water contamination (3). The areal ex-
tent of this at the water table as of June, 1959, is shown in Figure 15.
The figure illustrates also the ground water contours of the nearby
"mounds" and the relationship of the present (6) mounds to the distribu-
tion of contaminants. It must be emphasized, however, that the mounds

INDICATED DISTRIBUTION OF BETA - PARTICLE EMITTERS IN SOIL BENEATH 216-S-1&2 CRIBS
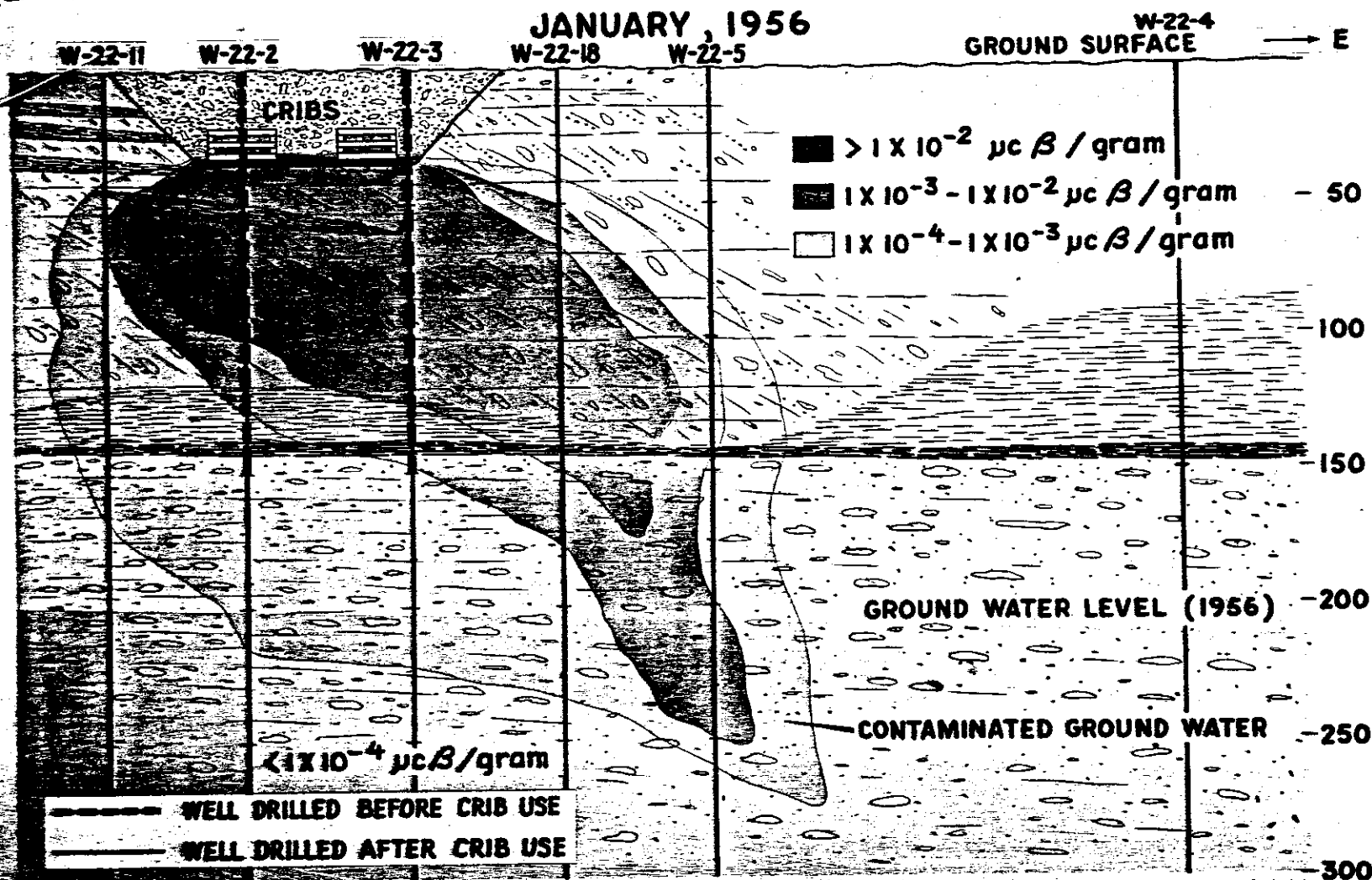
FIGURE 13

DISTRIBUTION OF RADIOISOTOPES BELOW S-1&2 CRIB

JANUARY, 1956

WELL DRILLED BEFORE CRIB USE.
WELL DRILLED AFTER CRIB USE.

FIGURE 14

MAP OF SEPARATIONS AREA, JUNE 1959

—400— GROUND WATER CONTOURS
         IN FEET ABOVE msl

  •   MONITORING WELLS

[mmmmm] 1.5 x 10^{-7}-1.0 x 10^{-6} μc BETA/cc

[:::::::] 1.0 x 10^{-6}-1.0 x 10^{-4} μc BETA/cc

[≡≡≡≡] >1.0 x 10^{-4} μc BETA/cc

MILES

KILOMETERS

N
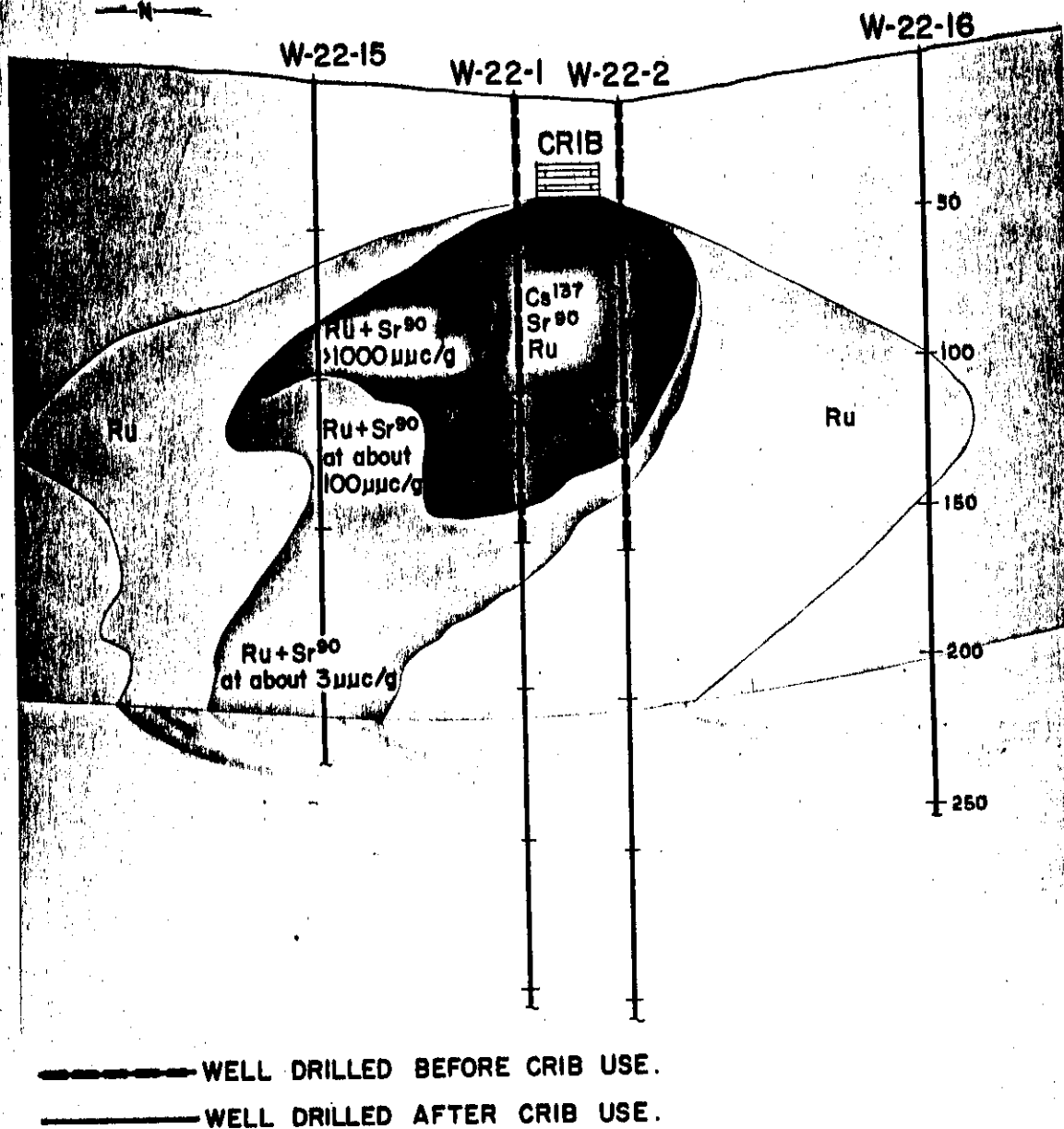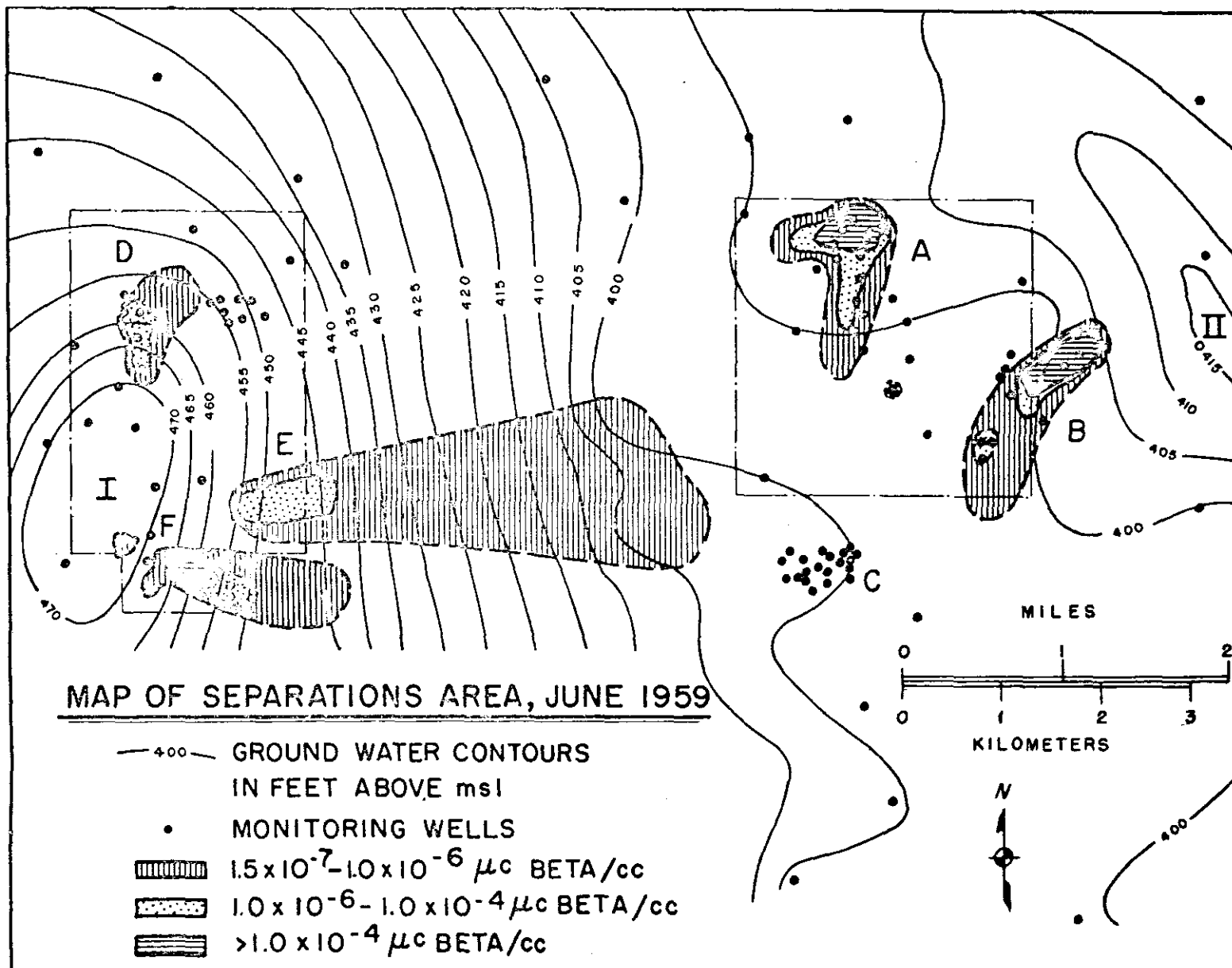
have increased and decreased in height during their history and have changed appreciably in shape and location. Conclusions as to the original source of the contamination in a given well must therefore be made with the full history of the disposals and the flow patterns in mind.

As can be seen on the map, there are six general areas of ground water contamination, the largest of which extends eastward from a point of disposal about 5 Km. The degree of ground water contamination is indicated. Some attention has been directed toward the sampling of well waters at depth and some locations have been found where contamination is greater at depth than at the water table or where it exists only at depth. These cases are under study; tentatively they are explained by the preferential carrying of solutions by more permeable strata leading to depth. The sinking of high salt wastes is very likely a contributing factor in some cases although the relative importance of such processes is not yet evaluated.

Concentrations range from as low as 1.5×10^{-7} uc beta emitters/cc at some distance away from the points of disposal to greater than 1.0×10^{-4} uc beta emitters/cc alongside the cribs; the maximum observed was 6.0×10^{-2} uc beta emitters/cc. Isotopic analyses are made on a schedule when concentrations of radioisotopes in routine ground water samples exceed 1×10^{-4} uc gross beta emitters/cc.

Most of the radioactive material in the ground water is ruthenium; however, small amounts of Cs^{137}, Co^{60} and Sr^{90} have also been detected. In 1956, Co^{60} and Cs^{137} were detected in the ground water (near A on the map) at distances up to 350 m from the disposal point. At that distance the concentrations were 1×10^{-5} and 1×10^{-6} uc/cc, respectively. At the present time, the Cs^{137} can no longer be detected and Co^{60} is found in seven wells immediately adjacent to the site, the maximum detected concentration of Co^{60} being 2.1×10^{-4} uc/cc.

Also in 1956, Cs^{137} was detected in a well 35 m from a crib site near point D at a concentration of 1.4×10^{-5} uc/cc. The Cs^{137} is no longer detectable at this site.

In April of 1957, Sr^{90} was detected in a well 5 m from a crib located in the area designated as point F at a concentration of 1.0×10^{-6} uc/cc. The Sr^{90} concentration is currently at a level of 1.7×10^{-6} uc/cc in the well immediately adjacent to this crib.

Table 2 lists the detection limits of the routine procedures currently used to analyze water samples from monitoring wells.

Table 2

Routine Analytical Detection Limits

Analysis	Sample Volume	Detection Limit
Gross Beta	100 cc	8×10^{-8} uc/cc
Uranium	0.1 cc	6×10^{-9} uc/cc
Co^{60}	100 cc	6×10^{-7} uc/cc
Sr^{90}	300 cc	7×10^{-8} uc/cc
Cs^{137}	25 cc	5×10^{-7} uc/cc

Much greater sensitivities can be obtained by special methods and individual handling of samples.

CONCLUSION

The radioactive waste disposal processes in use at Hanford have been supported over the years by closely correlated and balanced programs of theoretical study, laboratory and field experiment, and observational experience. Extensive research and development efforts in geology, hydrology, mineralogy, soil chemistry, analytical chemistry and process engineering have been applied to ground disposal problems. The effects of the large scale disposals illustrate that such operations may be conducted safely in a suitable environment.

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